Assessing the Health Benefits of Urban Air Pollution Reductions Associated with Climate Change Mitigation (2000–2020): Santiago, São Paulo, México City, and New York City

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To investigate the potential local health benefits of adopting greenhouse gas (GHG) mitigation policies, we develop scenarios of GHG mitigation for México City, México; Santiago, Chile; São Paulo, Brazil; and New York, New York, USA using air pollution health impact factors appropriate to each city. We estimate that the adoption of readily available technologies to lessen fossil fuel emissions over the next two decades in these four cities alone will reduce particulate matter and ozone and avoid approximately 64,000 (95% confidence interval [CI] 18,000–116,000) premature deaths (including infant deaths), 65,000 (95% CI 22,000–108,000) chronic bronchitis cases, and 37 million (95% CI 27–47 million) person-days of work loss or other restricted activity. These findings illustrate that GHG mitigation can provide considerable local air pollution–related public health benefits to countries that choose to abate GHG emissions by reducing fossil fuel combustion. *Key words:* air pollution, climate policy, greenhouse gases mitigation, morbidity, mortality, ozone, particulate matter, public health. — *Environ Health Perspect* 109(suppl 3):419–425 (2001). *http://ehpnet1.niehs.nih.gov/docs/2001/suppl-3/419-425cifuentes/abstract.html*

A broad-ranging and lively debate is under way about the long-term consequences of greenhouse gas (GHG)–emitting activities, such as fossil fuel combustion, for global climate change. Although it is generally understood that policies to reduce GHG emissions can also have near-term positive and negative ancillary side effects on public health, ecosystems, land use, and materials, these side effects have not been well characterized or integrated into policy analyses of mitigation (*1*). This article assesses near-term public health consequences of reductions in ambient concentrations of particulate matter (PM) and ozone (O_3) associated with policies to reduce GHG emissions.

The approach used in this work parallels that of other recent national and regional assessments in which estimates of public health impacts of air pollution are derived from epidemiology-based concentration–response functions. For example, Künzli et al. (*2*) relied on established coefficients of changes in health outcomes associated with increments of air pollution to estimate the impact of outdoor (total) and traffic-related air pollution on morbidity and mortality in Austria, France, and Switzerland. Air pollution was found to be associated with 6% of total mortality, or more than 40,000 attributable cases per year, with about half the mortalities linked to traffic-related emissions. Hall et al. (*3*) employed air pollutant mapping for the South Coast Basin of California and estimated that attaining ambient air pollution standards might save 1,600 lives per year in the region. The U.S. Environmental Protection Agency (U.S. EPA) employed a similar approach to assess the benefits of the U.S. Clean Air Act over its first two decades of application (*4*) and over the next two decades (*5*). The potential annual benefits from GHG mitigation options in the United States include thousands of avoided deaths and up to 520,000 work loss days that would result from the adoption of policies that target reductions in both air pollution and GHG emissions (*6*). Preliminary national assessments for Hungary and Canada have also suggested that substantial numbers of lives can be saved, chiefly from declines in mortality associated with fine particle reductions resulting from increased energy efficiency (*7,8*).

The Working Group on Fossil Fuels produced a global assessment that calculated the consequences of continuing energy policies that rely on customary practices, called business as usual (BAU), in 2010 and 2020. This study assessed the range of avoidable deaths solely from projected changes in PM that could arise between 2000 and 2020 under current policies and under the scenario proposed by the European Union in 1995 (*9,10*). On the basis of the estimated changes in PM and associated reductions in mortality, the report predicted that 700,000 avoidable deaths (90% confidence interval [CI] 385,000–1,034,000) will occur annually by 2020 under the BAU forecasts when compared with the climate policy scenario. As a first approximation, the cumulative impact from 2000 to 2020 on public health related to the difference in PM exposure could total 8 million deaths globally (90% CI 4.4–11.9 million). Thus, energy policies have significant impacts on air pollution and associated public health effects.

Scenario Development

As part of a collaborative project to promote consideration of public health impacts in the development of GHG mitigation policies in large cities today, this article develops scenarios that estimate the cumulative public health impacts of reducing GHG emissions from 2000 to 2020 in four cities: México City, México; São Paulo, Brazil; Santiago, Chile; and New York, New York. Using information from the published scientific literature, including those generated by local studies in each of these cities, we estimate what the adoption of climate policies would mean for public health by 2020 based on existing transportation and energy technologies that reduce the carbon intensity of fuels. For the four cities analyzed we estimate the baseline change in air pollutants associated with the adoption of energy efficiency and climate policies (CP) intended to reduce GHG emissions during the next 20 years. The CP scenario we apply to each city considers the attendant impact of GHG mitigation measures on the emissions of primary pollutants and consequent impacts on ambient concentrations of secondary pollutants. The scenarios used in this article are based largely upon a recent estimate of the potential reductions in PM and O_3 ambient

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concentrations that have been estimated as achievable in Chile through the use of readily available technologies to mitigate GHG emissions in energy, transport, residential, and industrial sectors (*11*). This Chilean study shows that the adoption of energy efficiency and fuel substitution measures in transportation, energy, residences, and industry can lower GHG emissions in 2020 by approximately 13% with respect to the BAU case, at little or no cost.

The associated reductions in ambient concentrations of PM less than 10 μm in diameter (PM_{10}) and O_3 are approximately 10% of the projected baseline levels in 2020 and occur gradually throughout the two decades. These results agree with another study for Chile developed independently by the Organisation for Economic Co-operation and Development (*12*). On the basis of these results, we compute the benefits associated with a 10% reduction in PM and O_3 that might be associated with climate change policies in each of these megacities. Although we recognize that other air pollutants may also affect human health, we consider only PM and O_3 in this analysis. These two presently have the best-documented health effects across a wide range of health outcomes and are pollutants that represent, respectively, the particulate and the gaseous components of air pollution. Therefore, although conservative in that only two pollutants are considered, this analysis does consider the breadth of types of air pollution and health effect changes that would result from GHG mitigation measures.

The magnitude and scale of potential benefits of GHG mitigation will vary with the stringency of existing and proposed regulations (*13*). Where baseline conditions include relatively high ambient levels of pollution and unsophisticated technology, as with many rapidly developing countries such as Chile, México, and Brazil, the potential benefits of reducing emissions will likely be much larger than where baseline conditions already include fairly strict regulatory controls and advanced systems. Mitigation policies in cities that have little regulatory control will have a greater absolute impact than those in areas with well-established controls in place. To develop the scenarios in this analysis, we made a number of simplifying assumptions. We assumed that the fuel and technology mixes in the Latin American cities we assessed here were similar. Although São Paulo relied on ethanol fuels until the mid-1990s, that fuel is not expected to play a major role in the future. On the basis of recent trend information, we assumed that the concentrations of both PM_{10} and O_3 would remain constant in these cities for the baseline scenarios. For Santiago, where progress is being made in the reduction of PM_{10}, we assumed a reduction of 3% per year in the concentrations of PM_{10}. Applying the 10% reduction to the projected concentrations for 2020 for México City, São Paulo, and Santiago, respectively, we project reductions of 6.4, 5.3, and 4.5 $\mu g/m^3$ in the annual average of PM_{10}, and 11.4, 6.5, and 4.9 ppb in the annual average of daily 1-hr maximum O_3 by 2020.

In the analysis of air pollution–related impacts associated with GHG mitigation for New York City, a 10% reduction in O_3 and PM_{10} from present baseline levels amounts to changes in the annual average concentrations of 3.9 ppb daily 1-hr maximum O_3 and 2.2 $\mu g/m^3$ annual average PM_{10}. This assumed GHG-associated reduction in air pollution compares well with the only estimate that has been made to date regarding potential reductions that could arise from GHG mitigation in the United States. Abt Associates (*6*) recently estimated for the U.S. EPA that a GHG policy of applying a fee of $56/ton of carbon emitted would be associated with a reduction of between 0.4 and 2.7 $\mu g/m^3$ $PM_{2.5}$ in the Midwest/Northeastern United States (or about 0.6 to 3.9 $\mu m/m^3$ PM_{10}, assuming 70% is $PM_{2.5}$) when compared with various baseline emissions scenarios. ($PM_{2.5}$ is the mass of suspended particles less than 2.5 μm in aerodynamic diameter that can penetrate to the deepest recesses of the lung.) Thus, although New York City is very different from Santiago, the assumption of a 10% reduction from present PM_{10} and O_3 levels in New York City appears reasonable. This is because this assumption is consistent with co-pollutant reductions previously projected to be associated with potential GHG mitigation measures in this region of the United States.

While the reductions in pollutant concentrations estimated here are similar between New York City and the Latin American cities, it must be noted that they might arise from very different policies. For developing countries, which are not required to abate GHGs under the Kyoto Protocol, the reductions stem from nonpositive cost measures of energy efficiency and fuel substitution, whereas for the United States, which is supposed to abate GHG emissions under the Kyoto Protocol, the reductions might result from the application of a carbon tax or other policies.

Health Effects Estimation Methods

To develop estimates of public health impacts in the cities of interest over the next two decades, we relied on published studies on air pollution and health, using concentration–response (C–R) coefficients derived from studies conducted in these or similar cities whenever possible. Table 1 shows a list of end points that have been associated with air pollution. The left side of the table displays effects associated with air pollution in numerous cities in more than 20 countries across the world; these effects are used extensively in standard setting (*4,5,14*). The effects in the right side of the table have been observed in some cities but have not yet been replicated in many locations. Assessments of air pollution–related health effects are based almost exclusively upon epidemiologic studies, which are corroborated by a body of knowledge from controlled human and animal studies (*5*). Most of the recent studies linking air pollution and health have applied multivariate methods, such as Poisson models, that address major potential confounders (*15*). In such models the expected value of the number of health effects is modeled as an exponential function of the explanatory variables, and the change in the number of health effects, *J*, when ambient concentrations of pollutant, *P*, change by ΔC_P, is given by

$$\Delta E_p^J = [\exp(\beta_p^J \cdot \Delta C_P) - 1] \cdot Pop^J \cdot BR_0^J.$$

β_p^J is often referred to as the C–R coefficient of end point, *J*, associated to pollutant *P*. BR_0^J is the base rate of effects, *J*, in the affected population Pop^J. For example, BR_0^J might be the number of deaths per 100,000 infants less than 1 year of age during a baseline year. However, it is important to note that the affected population is not necessarily the whole population of a city. It can be separated by age groups (e.g., infants, adults,

Table 1. Scope of human health effects of air pollution.[a]

Quantifiable health effects	Suspected health effects
Mortality (elder)	Induction of asthma
Mortality (infant)	Fetus/child developmental effects
Neonatal mortality	Increased airway responsiveness
Bronchitis: chronic and acute	Nonbronchitis chronic respiratory diseases
Increased asthma attacks	Cancer
Respiratory hospital admissions	Lung cancer
Cardiovascular hospital admissions	Behavioral effects (e.g., learning disabilities)
Emergency room visits for asthma	Neurologic disorders
Lower respiratory illness	Exacerbation of allergies
Upper respiratory illness	Altered host defense mechanisms (e.g., increased shortness of breath susceptibility to respiratory infection)
Respiratory symptoms	Respiratory cell damage
Days of work loss	Decreased time to onset of angina
Moderate or worse asthma status	Morphologic changes in the lung
Days with restricted activity	Cardiovascular arrhythmia

[a]Adapted from U.S. EPA (*5*).

elder) or by health conditions (e.g., asthmatic population) among many possible divisions.

Because β_P^J is small, the previous equation can be linearized and expressed in the following terms:

$$\Delta E_P^J = \left[\beta_P^J \cdot BR_0^J \cdot \frac{Pop^J}{Pop}\right] \cdot \Delta C_P \cdot Pop = IF_P^J \cdot \Delta C_P \cdot Pop,$$

where *Pop* is the total population of an area of analysis and IF_P^J is the health impact factor of pollutant *P* for end point *J*. It encompasses the relative risk, the base rate of the effect, and the relative size of the exposed population as a fraction of the total population. By expressing it this way, it is straightforward to compare the relative impact of pollution across different populations, and to compute the changes in health effects from the changes in concentrations and total population of a given city.

An important decision in the estimation of pollution effects involves the selection of pollutants for analysis. Incremental changes in mortality and morbidity associated with changes in exposures to PM have been documented extensively. A number of other common air pollutants, including sulfur dioxide, carbon monoxide, nitrogen oxides (NO_x), and O_3 have also been linked with various health effects. Inclusion of several pollutants in a single analysis simultaneously can lead to overestimation of the total effects if the C–R coefficients for each pollutant have been derived independently and the effects are added up later. Conversely, consideration of only one pollutant (PM_{10}, for example) to estimate the effects can underestimate the total air pollution effects, as several analyses show that the total risk of air pollution increases when more pollutants in addition to PM_{10}, such as O_3 and nitrogen dioxide (NO_2), are considered (*16,17*). In this analysis we considered two pollutants that have shown consistent and relatively independent associations with an array of adverse health effects: PM and O_3.

Ideally, the estimation of the change in health effects associated with concentrations in a given city should be based on studies conducted locally. However, local studies are not available for every health effect in every city. Therefore, we also use studies from other similar places, as available. For morbidity, C–R functions were derived from pooled estimates from the international literature when more than three studies were available. Random effects models were used to obtain a summary measure in this meta-analysis (*18*). For the case of O_3, studies taking into account the effects of PM_{10} simultaneously were employed when available. Latin American studies were preferred for the cities in that region. Regionally relevant information was available for the acute effects of PM on mortality for each of the cities. For example, whereas New York City infant mortality and hospital admissions effects were estimated using U.S.-based studies, the Latin American cities' pollution effect on infant mortality was derived from México City (*19*), child medical visits from Santiago (*20*), and hospital admissions from São Paulo (*21*).

The effects of PM exposure on mortality have been derived both from time–series studies that evaluate the effect of several days of elevated or acute exposures on daily mortality and from cohort studies that evaluate the annual changes in mortality or morbidity associated with long-term, chronic exposures. Because the estimates from these two types of studies focus on different time frames, we used both types to indicate the likely upper- and lower-bound effects on mortality. Two main cohort studies of the general population have reported different central effect estimates, although their CIs overlap. We used the Pope et al. (*22*) and the Dockery et al. (*23*) studies to derive the central and high estimates of mortality, respectively. A recent reanalysis of these studies has been conducted by the Health Effects Institute and has confirmed their results (*24*). The lower bound of mortality effects were obtained from time-series studies, as shown in Table 2. Some have raised the issue that these studies may reflect life shortening of only a few days or weeks [sometimes referred to as "harvesting" (*25*)], but recent analyses have not borne out this assertion (*26–28*). The effects of O_3 on mortality in the Latin American cities were obtained from a meta-analysis of several studies.

In New York, the health effects associated with the GHG mitigation–induced air pollution decreases were based largely on the C–R coefficients derived from the published literature for New York State by the Empire State Electric Energy Research Corporation (ESEERCO) as part of the New York State Environmental Externalities Cost Study (*29*). In some cases, however, alternative study results were used to define the bounds, such as in the case of PM_{10}-associated mortality. In this case, as for the Latin American cities, the Pope et al. study (*22*) of the effects of chronic PM exposure was used as the central estimate, the Dockery et al. study (*23*) was used to estimate the upper bound, and the results of a time–series PM study of the effects of acute PM exposure were used for the lower bound (as derived from the ESEERCO report). In addition, the ESEERCO C–R coefficients were updated

Table 2. Studies considered in the analysis of the mortality effects of PM_{10} for México City, São Paulo, and Santiago.

Estimate	City	Age group	Relative risk for 10 μg/m³ increase in PM_{10} (95% CI) (percent)	Reference
All ages mortality				
Low estimate	México City	All	1.2 (1.1–1.2)	(*33*)
	São Paulo	All	0.85 (0.46–1.2)	(*34,35*)
	Santiago	All	0.7 (0.3–1.0)	(*16*)[a]
Mid estimate	All cities	> 30 years	3.5 (1.9–5.2)	(*22*)[a]
High estimate	All cities	> 25 years	6.8 (2.7–11.4)	(*23*)[a]
Infant mortality	All cities	< 1 year	4.0 (1.6–6.5)	(*19*)

[a]Relative risks were multiplied by 0.55 when the pollutant in the original study was $PM_{2.5}$.

Table 3. O_3 and PM_{10} health impact factors and CIs for New York City.

Health effect outcome	Health impact factor per million inhabitants: Mid	95% CI	Reference
Ozone impacts (effects per part per billion of annual average daily 1-hr maximum ozone)			
Acute mortality	1.2	(0.0–2.4)	(*36*)
Acute respiratory hospital admissions	5	(3–7)	(*37*)
Acute emergency department visits	40	(25–55)	(*38*)
Acute asthma attacks	1,005	(570–2,747)	(*39*)
Acute restricted activity days	17,000	(7,000–27,000)	(*40,41*)
Acute respiratory symptom days	50,000	(26,000–78,000)	(*42*)
PM_{10} impacts (effects per microgram per cubic meter of PM_{10})			
Acute and chronic infant mortality	0.21	(0.1–0.3)	(*43*)
Acute and chronic adult mortality	33	(8–52)	(*22*)
Acute respiratory hospital admissions	12	(7–17)	(*44*)
Chronic adult bronchitis	39	(19–59)	(*45*)
Acute bronchitis in children	53	(26–78)	(*46*)
Acute emergency department visits	94	(55–130)	(*38*)
Acute asthma attacks	774	(446–2,589)	(*39*)
Acute work loss days	5,300	(2,700–8,300)	(*47*)
Acute restricted activity days	14,900	(7,616–23,509)	(*48*)
Acute respiratory symptom days	170,000	(81,000–259,000)	(*42*)

with newer studies as appropriate, and studies conducted in or near New York State were chosen when available.

For each end point, we derived health impact factors (as defined above) specific for each city, as presented in Tables 3–5. Although some of the relative risks are similar for the cities, the impact factors differ because the population distribution of those affected differs between cities, as does the baseline rate of hospitalization and other end points. For instance, infant mortality impacts are higher in Latin American cities, in part because they have higher baseline rates. Conversely, the impact factors for adult health outcomes (e.g., work loss days and chronic bronchitis) are usually higher per million people for New York City than for the Latin American cities, largely due to the higher percentage of more frail, older individuals in this more-developed, lower birth rate city. In addition, this analysis assumes a higher percentage of the PM_{10} is $PM_{2.5}$ in New York City versus the developing nation cities (70% vs 55%), which increases the impacts per $\mu g/m^3$ of PM_{10} reduction in New York City versus the other cities. To estimate the total health effects avoided during the next two decades, we applied the impact factors to the projected population and concentration changes, making the simplifying assumption that the current health conditions in these cities would prevail and that population would increase according to official projections, except in New York City, where a stable population size was assumed.

Table 4. PM_{10} health impact factors (with 95% CIs) developed for México City, São Paulo, and Santiago.[a]

End point	Age group (years)	México City Mid	México City 95% CI	São Paulo Mid	São Paulo 95% CI	Santiago Mid	Santiago 95% CI	References
Mortality effects								
Mortality	All[b]	21	(7–41)	20	(2–39)	12	(2–23)	See Table 2
Infant mortality	< 1	2.2	(0.9–3.6)	1.2	(0.5–2.0)	0.9	(0.4–1.5)	See Table 2
Chronic morbidity effects								
Chronic bronchitis	> 30	26	25.6	21.1	(8–43)	20.9	(6–36)	(*49*)
Hospital admissions								
Cardiovascular causes	All	2.4	(2–3)	2.4	(2–3)	2.4	(2–3)	Pooled from Los Angeles (*50*), Ontario (*51*), Detroit (*52*), Michigan (*53*), Chicago (*54*)
Respiratory causes	All	5.7	(4–8)	5.7	(4–8)	5.7	(4–8)	Pooled from California (*55*), Ontario (*51*), Paris (*56*), London (*57*), Amsterdam (*58*), Rotterdam (*58*), Cleveland (*59*), Buffalo (*37*), New York (*37*), Ontario (*60*), Milan (*61*), Los Angeles (*50*)
Children: all causes	< 13	3.2	(2–5)	8.2	(4–12)	8.2	(4–12)	São Paulo (*21*)
Other hospital visits								
Emergency room visits	All	141	(0–351)	93	(0–232)	93	(0–232)	Pooled from London (*62*), México (*63*), Quebec (*64*), Steubenville (*65*)
Children: medical visits	3–15	63	(14–112)	155	(34–275)	83	(18–147)	Santiago (*20*)
Morbidity effects								
Asthma attacks	All	668	(161–1,175)	668	(161–1,175)	668	(161–1,175)	(*39*)
Children: acute bronchitis	8–12	65	(0–145)	64	(0–143)	62	(0–138)	(*46*)
Restricted activity								
Work loss days	18–65	3,655	(3,095–4,216)	3,965	(3,357–4,573)	3,657	(3,096–4,217)	(*47*)
Restricted activity days	18–65	10,539	(9,287–11,792)	11,431	(10,073–12,790)	10,543	(9,291–11,796)	(*47*)

[a]Effects per million people per microgram per cubic meter PM_{10}. [b]The age groups considered vary according to the estimate (see Table 2 for details).

Table 5. Ozone health impact factors and 95% CIs developed for México City, São Paulo, and Santiago.[a]

End point	Age group (years)	México City Mid	México City 95% CI	São Paulo Mid	São Paulo 95% CI	Santiago Mid	Santiago 95% CI	References
Acute mortality								
Total	All	1.9	(1.0–2.7)	1.8	(1.0–2.7)	1.0	(0.2–1.7)	México City and São Paulo: pooled from London (*66*), México (*67*), México (*33*), Chicago (*68*), Philadelphia (*69,70*), Amsterdam (*71*), European cities (*72*), Rotterdam (*73*). Santiago: summer (*16*)
Hospital admissions								
Respiratory causes	All	15	(2–29)	15	(2–29)	15	(2–29)	Pooled from Buffalo (*37*), New York (*37*), Ontario (*60*)
All causes: children	< 5	0.29	(0.03–0.55)	0.65	(0.07–1.23)	0.50	(0.05–0.94)	Brazil (*74*)
Other hospital visits								
ERV for RSP causes	All	144	(76–212)	95	(50–140)	144	(76–212)	Pooled from Montreal (*64*), México (*75*), Ontario (*17*)
Morbidity effects								
Asthma attacks	All	1,140	(0–2,745)	1,140	(0–2,745)	1,140	(0–2,745)	(*46*)
Restricted activity days	Adults	11,781	(8,036–13,678)	12,779	(8,716–14,835)	11,786	(8,039–13,683)	

ERV for RSP, emergency room visits for respiratory causes.
[a]Effects per million people per part per billion per cubic meter annual average daily 1-hr maximum O_3.

Results

Tables 6 and 7 present the health effects estimated to be avoided in the four cities for the period 2000–2020 if modest GHG mitigation measures are initiated now, due to expected reductions in concentrations of PM and O_3, respectively. Figure 1 shows the total effects due to PM and O_3 reductions for all four cities combined. Not all end points could be evaluated for all cities, as relevant studies were not always available for every outcome for every city. Therefore, some of the estimations presented here are likely to be underestimates.

For the period analyzed, nearly 64,000 (95% CI 18,000–116,000) premature deaths associated with air pollution could be avoided if policies aimed at achieving modest reductions in GHG emissions are undertaken now. We also estimate that at least 65,000 (95% CI 22,000–108,000) new cases of chronic bronchitis; 91,000 (95% CI 28,000–153,000) hospital admissions; and 787,000 (95% CI 136,000–1,430,000) emergency room visits could be avoided. Other effects may appear less important, but their numbers are quite high. We predict that 6.1 million (95% CI 0.49–14.1 million) asthma attacks can be avoided, as well 37 million (95% CI 27–47 million) work loss days or other days with restricted activity.

The effects in children are also important to note. Premature deaths of infants less than 1 year of age (including neonatal mortality) that can be avoided amount to 4,100 (95% CI 1,600–6,700). In addition, more than 161,000 (95% CI 0–350,000) cases of acute bronchitis in children 8–12 years of age as well as 202,000 (95% CI 45,000-360,000) medical visits of children 3–15 years of age could be avoided. (This last effect was evaluated only in the Latin American cities.)

Discussion

Analyzing the total burden on human health from ambient air pollution in a community remains challenging, given the uneven nature of information on which such assessments must draw, the absence of information on many key pollutants, and the wide range of uncertainties characterizing many parts of the process. Because we have data from the cities reviewed here on the entire age range of the population and a broad range of effects that could occur throughout this range, this analysis provides a more robust assessment than past global efforts (*10*). Still, there is much that is not addressed in this work, including impacts on ecosystems, water supply and quality, agriculture, and materials damage.

We believe that our analysis is conservative for various reasons. First, we assumed no major changes in technologies for transport, energy, and commerce in the next two decades. Some air pollutants (such as O_3, some aromatic hydrocarbons, and NO_2) continue to increase in many metropolitan regions; their full impact is not assessed in this analysis. In addition, we could not include estimates of synergistic effects between various air pollutants, or with cofactors such as pollen and other allergens. New studies have indicated that synergies can occur between air pollutants and allergens. Thus, for example, British analyses show that physician visits for asthma and allergic rhinitis are increased more than additively when both pollen and air pollution levels are elevated (*30*). We also did not consider effects tied with cancer (*31*) and other diseases linked with exposures to pollution and other airborne toxics, which are not usually monitored and can be quite high in rapidly developing areas (*32*).

The rapid pace of urbanization globally means that more people are living in large cities than at any point in human history. Policies that increase energy efficiency and

Table 6. Health effects avoided from 2000 to 2020 in the four cities due to PM_{10} reductions if GHG mitigation measures are taken.

	Age group	México City		São Paulo		Santiago		New York	
End point	(years)	Mid	95% CI	Mid	95% CI	Mid	95% CI	Mid	95% CI
Mortality effects									
Mortality	All	29,055	(9,265–56,293)	11,225	(1,173–21,749)	3,960	(801–7,673)	8,785	(2,130–13,842)
Infant mortality	< 1	3,065	(1,187–4,944)	701	(271–1,130)	320	(124–516)	56	(43–75)
Chronic morbidity effects									
Chronic bronchitis	> 30	35,353	(10,904–59,803)	11,899	(3,670–20,129)	7,087	(2,186–11,989)	10,382	(5,058–15,706)
Hospital admissions									
Cardiovascular causes	All	3,341	(2,339–4,344)	1,365	(956–1,775)	819	(575–1,062)		
Respiratory causes	All	7,894	(5,026–10,763)	3,226	(2,054–4,398)	1,934	(1,231–2,637)	3,194	(1,863–4,525)
Children: all causes	< 13	4,475	(2,282–6,668)	4,638	(2,365–6,911)	2,788	(1,422–4,154)		
Other hospital visits									
Emergency room visits	All	195,355	(0–484,682)	52,756	(0 –130,890)	31,631	(0 –78,478)	25,023	(14,641–34,606)
Medical visits	3–15	87,064	(19,217–154,912)	87,377	(19,286–155,468)	28,054	(6,192–49,916)		(0–0)
Morbidity effects									
Asthma attacks	All	923,315	(222,624–1,624,005)	377,312	(90,975–663,649)	226,226	(54,546–397,906)	205,988	(118,839–689,268)
Acute bronchitis	8–12	89,897	(0–200,805)	36,157	(0 –80,764)	20,876	(0 –46,631)	14,055	(6,838–20,893)
Restricted activity									
Work loss days	18–65	5,051,218	(4,276,788–5,825,648)	2,238,907	(1,895,648–2,582,167)	1,238,111	(1,048,289–1,427,933)	1,404,099	(717,650–2,215,355)
Restricted activity days	18–65	14,563,826	(12,833,091–16,294,560)	6,455,286	(5,688,153–7,222,418)	3,569,759	(3,145,537–3,993,982)	3,966,380	(2,027,261–6,258,066)

Table 7. Health effects avoided from 2000 to 2020 in the four cities due to ozone reductions if GHG mitigation measures are taken.

	Age group	México City		São Paulo		Santiago		New York	
End point	(years)	Mid	95% CI	Mid	95% CI	Mid	95% CI	Mid	95% CI
Acute mortality									
Total	All	4,610	(2,500–6,720)	1,256	(681–1,831)	293	(54–533)	566	(0–1,133)
Hospital admissions									
Respiratory causes	All	37,964	(4,746–71,183)	10,588	(1,324–19,853)	4,738	(592–8,884)	2,360	(1,463–3,256)
All causes	< 5	721	(79–1,363)	447	(49–846)	153	(17–289)		
Other hospital visits									
ERV for RSP causes	All	354,005	(186,495–521,515)	65,245	(34,372–96,119)	44,183	(23,276–65,090)	18,876	(11,798–25,955)
Morbidity effects									
Asthma attacks	All	2,800,570	(0–6,744,230)	781,070	(0–1,880,943)	349,538	(0–841,745)	474,260	(268,747–1,296,309)
Restricted activity days	Adults	28,942,806	(19,742,221–33,601,432)	8,755,334	(5,972,114–10,164,590)	3,613,755	(2,464,984–4,195,424)	8,022,300	(30,303,300–12,741,300)

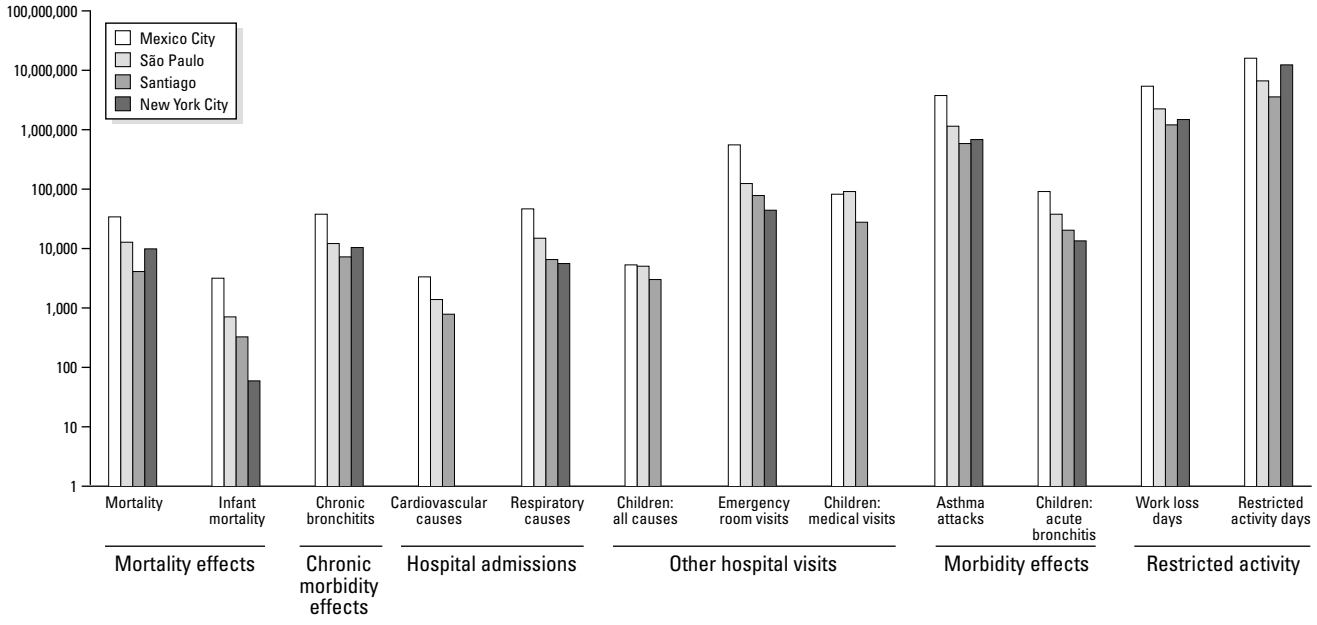

Figure 1. Health effects avoided from 2000 to 2020 in the four cities analyzed if GHG mitigation measures are taken.

promote less carbon-intensive fuels can yield a broad array of benefits by simultaneously improving local and regional air pollution and reducing the long-term buildup of GHG. Given the current and projected patterns of population growth and air pollution concentrations, the cumulative potential impact on public health from fossil fuels for the next two decades is quite high for any of the cities analyzed. Part of this burden can be avoided if GHG abatement policies to reduce the net use of fossil fuels are adopted. For Santiago and the other cities, the adoption of currently available technologies in energy, transport, residences, and industry can reduce population exposure to air pollution by at least 10% by 2020, yielding the associated public health benefits we have presented. Similar types of health impact reductions are expected to occur in other developing world cities if GHG emissions mitigation policies are also implemented in those cities.

This current collaborative analysis stems from an effort to integrate into the public discussion of GHG mitigation policies a consideration of what these policies might mean for public health in the nearer term. We fully recognize that population patterns can be changed by events not integrated into this assessment. We also are not unaware of promising technologies, such as fuel cells for energy production and transportation applications, that might substantially alter air pollutant emissions in the future. The estimates developed here are offered to illustrate the potential scale and scope of impacts. A fuller accounting would likely show still greater public health and natural resource benefits than this preliminary assessment.

Conclusion

The estimates offered here are presented for three purposes: to stimulate discussion of the full range of potential impacts of changes in future patterns of air pollution in developed and developing countries; to bring the subject of public health benefits to the attention of those who are formulating public policy regarding GHG mitigation; and to encourage additional research on this topic. The preliminary results in this work indicate that policies aimed at mitigating GHG emissions can provide a broad range of more immediate air pollution benefits to public health in the countries that implement these GHG mitigation measures. Conversely, not acting or postponing actions to reduce GHG emissions will fail to achieve the health benefits we present in this analysis.

The estimates of the potential public health benefits from the adoption of GHG mitigation policies offered in this paper were developed opportunistically, incorporating results from a variety of studies conducted for different purposes. While it must be acknowledged that there is still much that remains to be learned about the intricacies of climate change and the respective (and potentially interactive) roles of the various air pollutants, the numbers developed for these four cities illustrate the magnitude and scale of the air pollution impacts that may be averted by implementing GHG mitigation measures, according to current understanding. As pressures mount for actions to be taken to reduce GHG emissions, decisions being made in the next decade will affect the forms of energy production and transportation systems that will fuel this century in many regions. Further, efforts to promote a sounder accounting of how these technologies will affect public health must be encouraged. The failure to provide a fuller tally of potential health damages tied with air pollution in the discussions of GHG mitigation policies not only limits the utility of those discussions but also fails to meet a basic human concern. People care greatly about their health and the health of their children. Decision makers need to be well informed about the extent to which global climate policies adopted today can be expected to affect public health in both the near and long term.

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